area in which the air pollution models predict relatively high facility-related impacts. The four lakes/ponds that best satisfied these criteria are listed in Table 4.2.

The modeled air dispersion concentrations at the four lakes/ponds are also listed in Table 4.2. These values are constructed by weighting the modeled impacts of several receptors of the polar coordinate grid that fall within the pond/lake's watershed. Two values are determined for each water body: a concentration over the surface of the water body, and a concentration over lands within the watershed. Table 4.3 lists the modeled concentrations at the specific receptors that are used to construct the average impacts presented in Table 4.2. Average transfer coefficients are calculated by:

$$\alpha_{c,avg} = \frac{\sum\limits^{n} \alpha_c f_c}{\sum\limits^{n} f_c} \tag{4.2}$$

where the terms are:

$\alpha_{c,avg}$ the weighted-average transfer factor ($\mu g/m^3$ per g/s);

α_c the air dispersion transfer factor at the individual receptor ($\mu g/m^3$ per g/s), as described in Equation (4.1);

f_c the weighting factor for an individual receptor; and

n the number of receptors in the watershed.

4.1.3 Pollutant-specific concentrations in air and rates of deposition

The modeled air pollutant concentrations in Section 4.1.2 are generated with a nominal emission rate of 1 g/s. Estimates of specific pollutant concentrations in air at the point of maximum impact, as predicted by Equation (4.1) using individual pollutant emission rates (Table 4.4) and the α_c values discussed in Section 4.1.2, are presented in Table 4.5.

Pollutants are likely to be emitted from the NEPERA incinerator in both vapor and particle-bound phases. The relative volatilities of the chemicals of concern suggest the consideration of two broad categories. Volatile compounds, such as the organic waste stream constituents and inorganic compounds such as ammonia and possibly some metals, are likely to be released as vapors. Metallic compounds such as nickel, chromium, and lead, however, may condense onto or within solid particles.

For this assessment we treat organic compounds and non-metallic inorganic compounds as vapors, and metallic compounds as particle-bound pollutants. The significance of this

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generalization pertains to the assumed treatment of atmospheric deposition. Specifically, the following assumptions are made:

- chemicals emitted as vapors are assumed not to deposit to an appreciable extent within the study domain;[5] and

- metallic compounds released from the stack are assumed to deposit at ground-level to water, soil, and vegetation as components of small particles.[6]

The deposition rate D_p (mass per unit area per unit time) is estimated as:

$$D_p = c_a v_d \tag{4.3}$$

where v_d is a deposition, or settling, velocity of the airborne particles. Deposition velocities, which can differ among chemical species, are typically determined using procedures published by the California Air Resources Board (CARB) or other sources for the assessment of deposition rates of aerosol emissions from stationary sources. In the CARB procedure, Equation (4.3) may be applied on an hourly basis, and the deposition velocity estimated from local meteorological conditions (wind speed, atmospheric stability, and temperature), local terrain, and particle characteristics (density and size distribution). Hourly depositions are summed for each year of the multi-year modeling period to obtain average estimates of annual deposition rates of contaminants that may be emitted from the incinerator stack. Since the air concentration (c_a) is proportional to emission rate (E), time-averaged annual particle deposition rates D_{pt} may be expressed as the product of the contaminant emission rate (E) and a normalized deposition parameter β_d :

$$D_{pt} = E\beta_d \tag{4.4}$$

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[5] The tendency for these pollutants not to bioaccumulate in environmental media is further justification for assuming insignificant deposition.

[6] Small particles are expected during typical operation because metals are not used in NEPERA production processes and not intentionally burned in the incinerator — the large residual particles that could be expected from the burning of metal-bearing wastes are likely to be absent from the flue gas. Hence, metals from the flue gas are expected to create or condense onto small particles. In previous stack testing of the facility, the fact that particle loadings were difficult to collect in significant quantities supports this assumption (based on conversation with Martin, 1994).

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In addition to meteorological parameters, the normalized deposition rate (β_d) of a particle-bound contaminant depends on particle properties. Figure 4.5 depicts deposition velocities v_d estimated with the CARB procedures for various particle sizes and atmospheric roughness heights. Four curves correspond to roughness heights ranging from 5 cm to 100 cm, which are characteristic of land use ranging from relatively smooth surfaces such as water to well-vegetated areas such as forests. Values at each point are calculated by weighting deposition velocities over the spectrum of wind speeds and stability classes present within the meteorological data collected at Stewart Air Force Base.

Assuming the NEPERA facility emits particles in the 0.1–1.0 μm size range, deposition velocities can be expected to be less than 0.1 cm/s. As a simplification, a single deposition velocity v_d of 0.1 cm/s is assumed to estimate deposition rates[7]. The normalized deposition rates at the location of maximum impact and within the four watershed areas are provided in Table 4.6. Table 4.7 lists the compound-specific deposition rates predicted by Equation (4.4) for the maximum impact point and the two watershed areas (Cranberry Lake and Swimming Pond) used to assess exposure.[8]

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[7] Particulate-bound compounds emitted by the NEPERA facility may either be adsorbed to the surface of a particle or entrained in the particle. Surface-weighted and mass-weighted deposition velocities are usually used for each of these cases, respectively. However, since the particle size distribution in the stack gas is unknown, surface-weighted and mass-weighted deposition velocities cannot be calculated with precision.

[8] As explained in Section 2.2, only two of the four lakes and ponds evaluated enter into the quantitative risk assessment. These are Cranberry Lake and Swimming Pond. Although the deposition rates to the other waterbodies evaluated are higher than those to Cranberry Lake and Swimming Pond, the surface water modeling (described in Section 4.3) predicts higher overall impacts to Cranberry Lake and Swimming Pond as a result of differences in other modeling parameters such as soil runoff and surface area of the waterbody. The parameters listed in Table 4.6 and elsewhere allow for the calculation of surface water concentrations in the other two lakes.
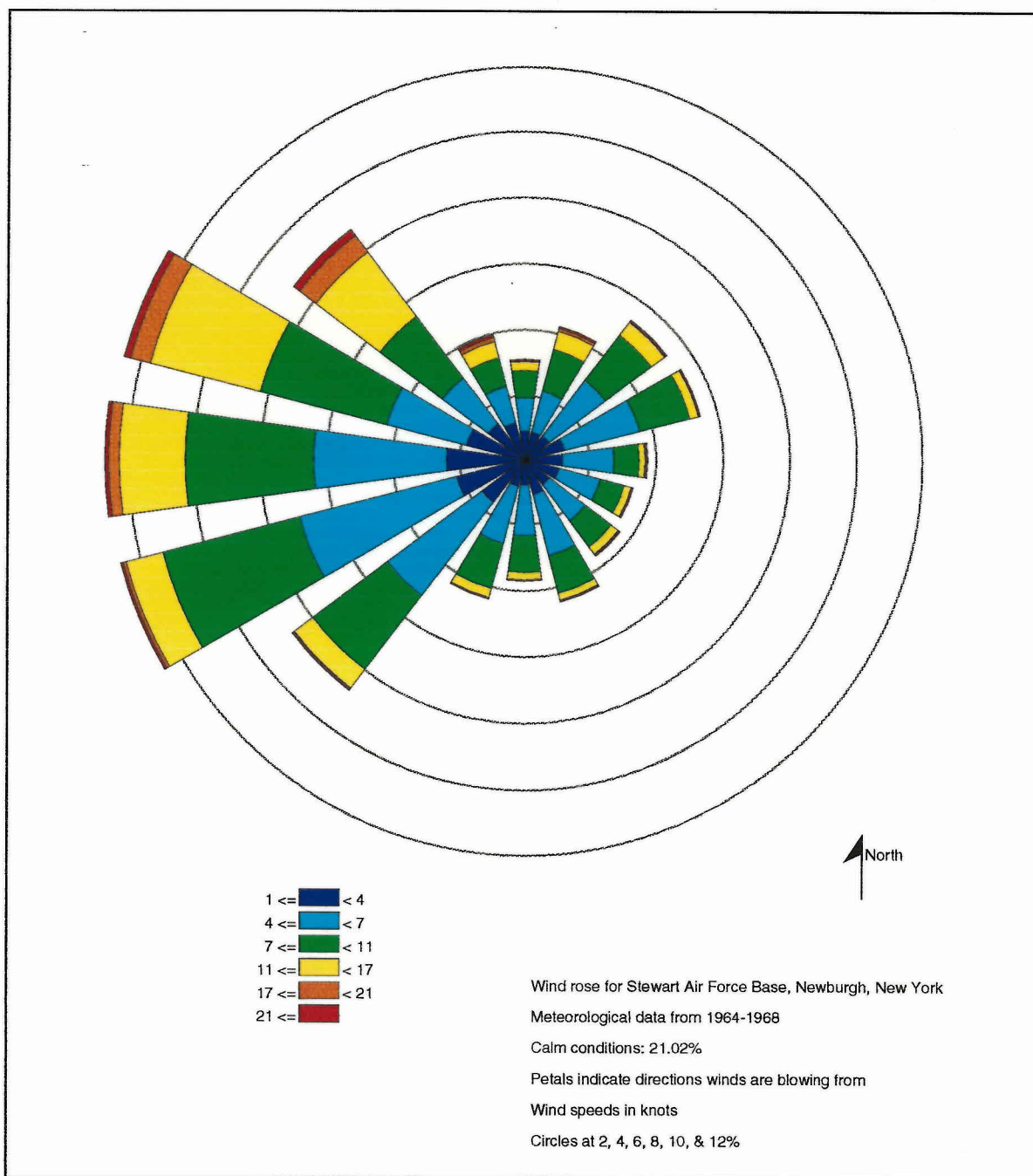
4-7

Figure 4.1   Wind rose of meteorological data collected at the Stewart Air Force Base, Newburgh, New York, from 1964–1969. Petals indicate the direction from which winds originate, and the length of each petal indicates the frequency of occurrence. The colors of each petal reflect different wind speed categories.
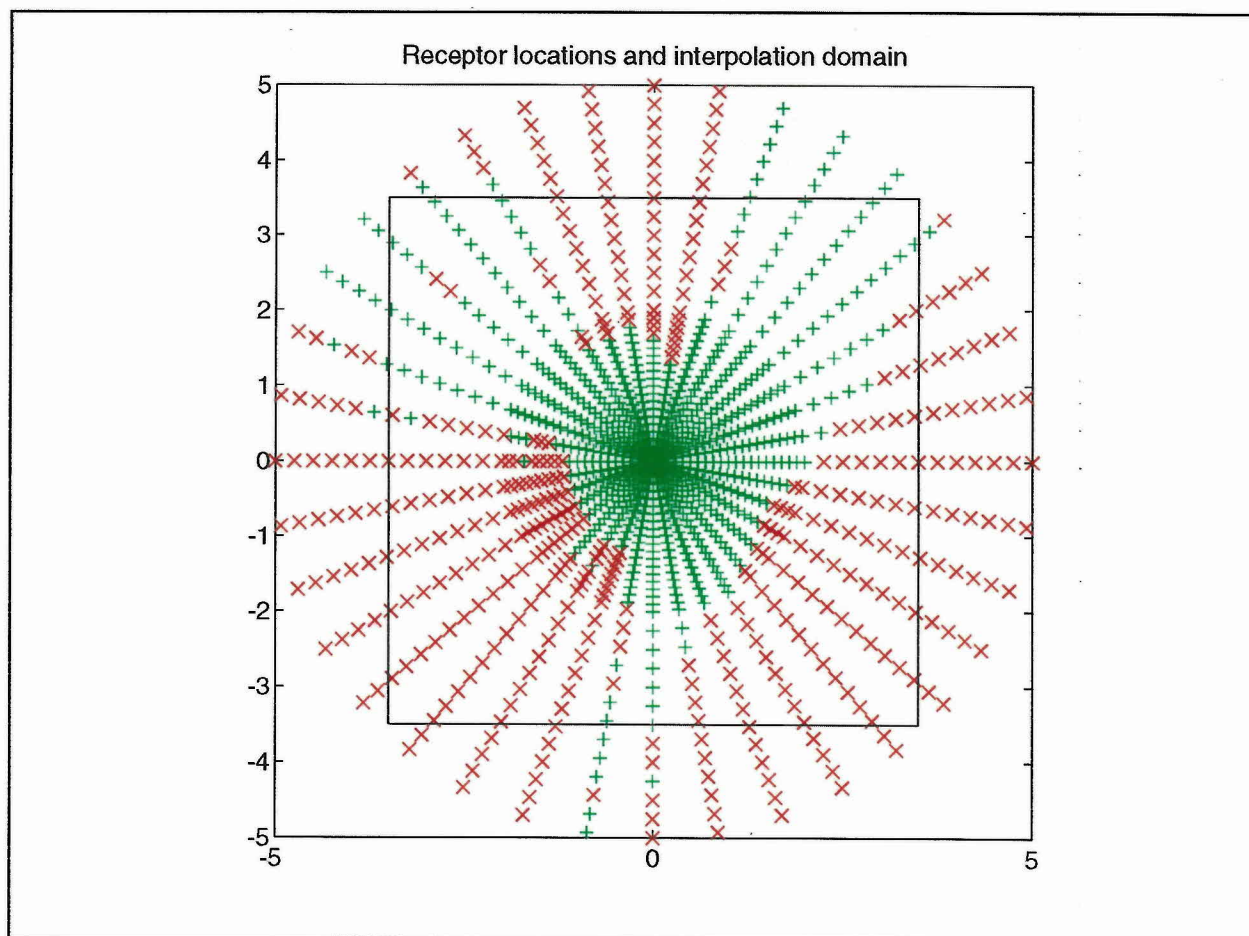
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Receptor locations and interpolation domain

Figure 4.2   Configuration of the polar coordinate modeling domain. Receptor locations indicated in green are below stack-top and evaluated with the ISCST2 model. With a stack base elevation of 532 feet amsl and a stack height of 152 feet, the stack-top is at 684 feet amsl. Receptors shown in green are thus at or below 680 feet amsl. (Receptors are reported to the nearest 10 feet). Receptor locations indicated in red are above stack-top — at or above 690 feet amsl. These receptors are predominantly evaluated with the COMPLEX1 model. Scales on the x and y axes indicate km from the center of the stack, which is located at the origin (0, 0). The boxed area at ±3.5 km from the stack delineates the area of analysis for Figure 4.3 and Figure 4.4.

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Interpolated terrain elevations (ft amsl)
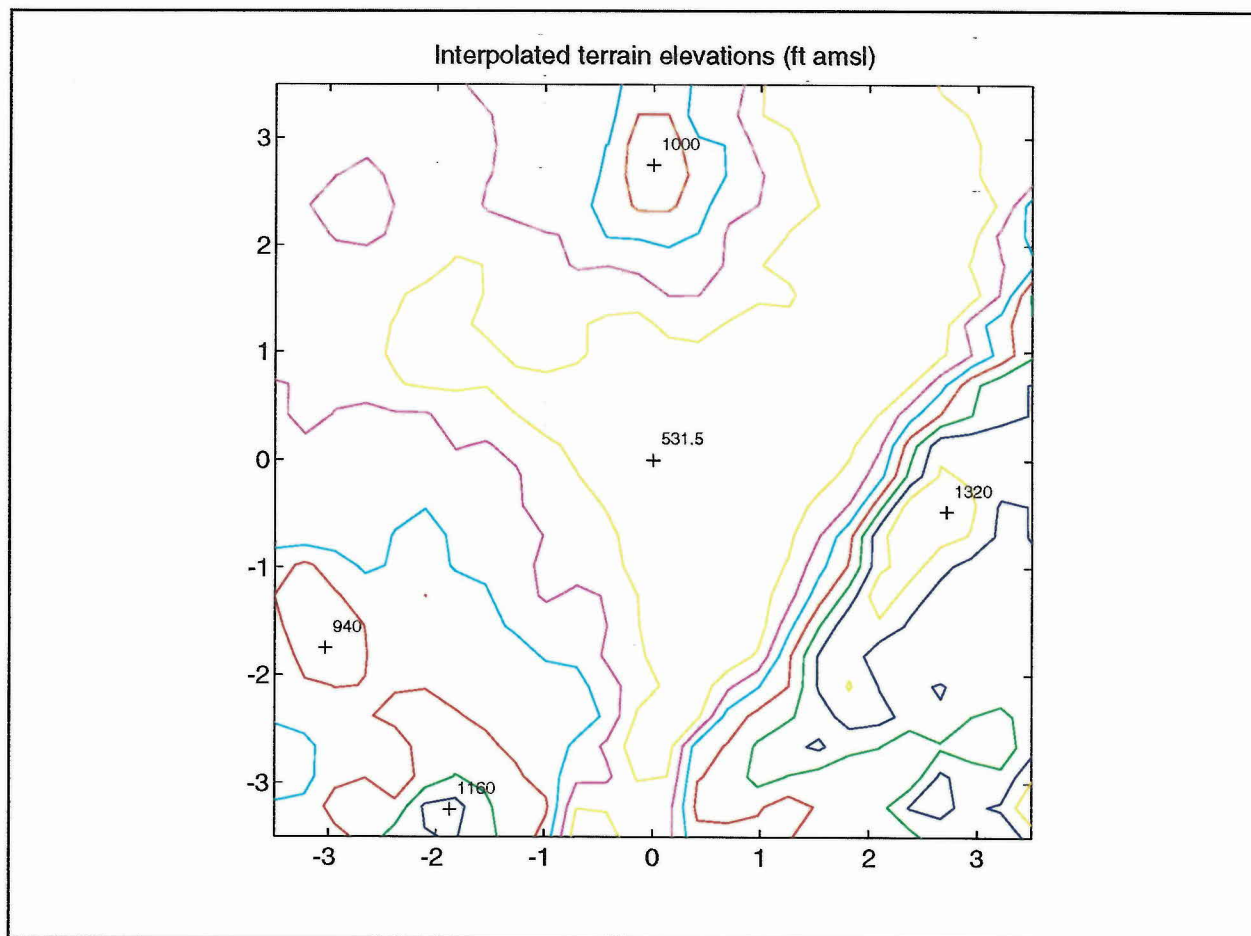
Figure 4.3 Terrain elevation contours in a 7×7 km^2 portion of the study area. Contour levels, in feet above mean sea level (amsl), are 600 (yellow), 700 (magenta), 800 (light blue), 900 (red), 1,000 (green), 1,100 (dark blue), and 1,200 (yellow again). Scales on the x and y axes indicate km from the center of the stack, which is located at the origin (0, 0). Terrain contours are constructed from a weighted rectangular grid and tend to underestimate peak values, as explained in the text in footnote 3. For reference, actual (non-gridded) terrain elevations at the stack location and at local maxima are indicated.

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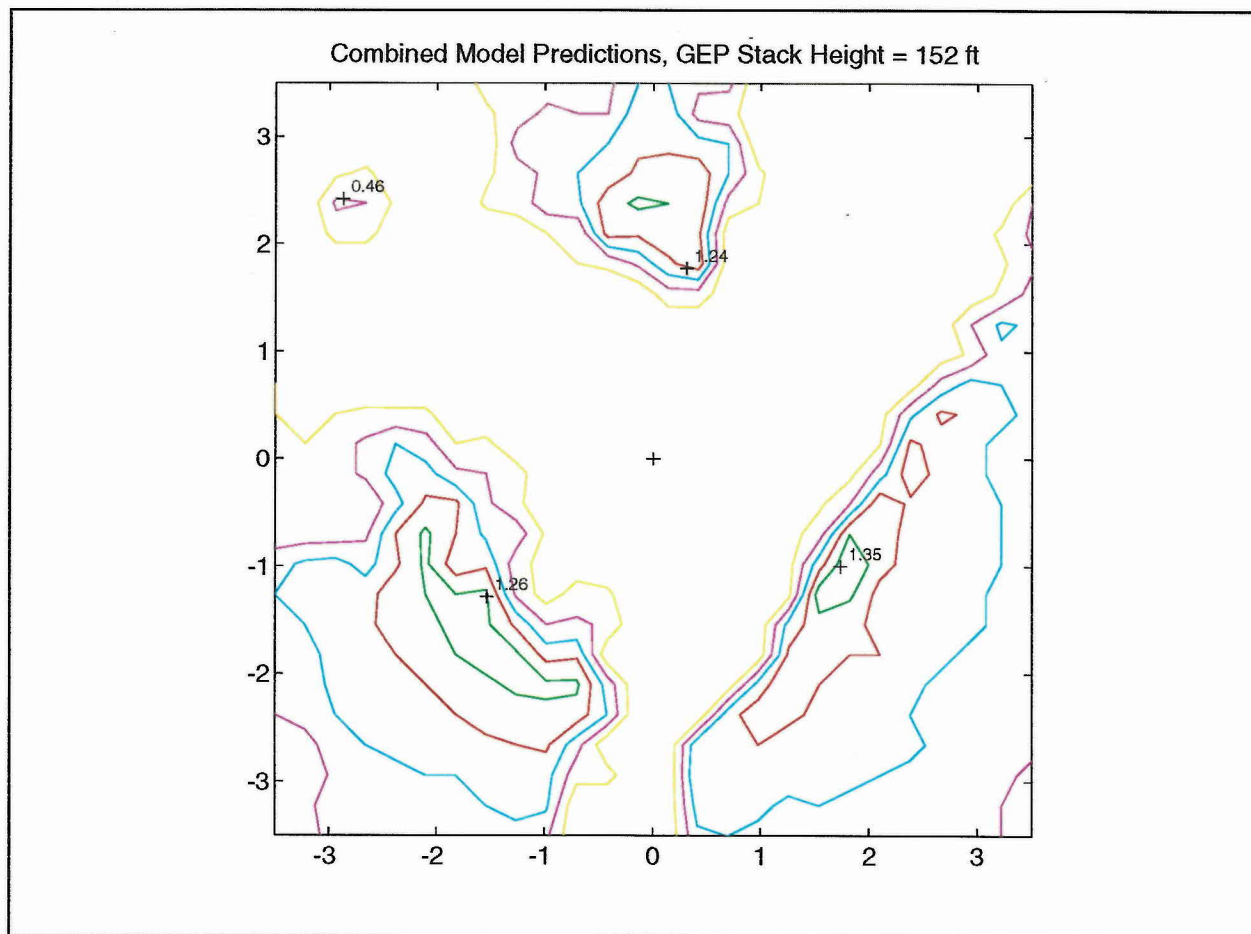
Figure 4.4   Contours of modeled air pollutant concentrations in a 7×7 km^2 portion of the study area. Contour levels, in $\mu g/m^3$ per g/s, are 0.2 (yellow), 0.4 (magenta), 0.6 (light blue), 0.8 (red), and 1.0 (green). Scales on the *x* and *y* axes indicate km from the center of the stack, which is located at the origin (0, 0). Contours are constructed from a weighted rectangular grid and tend to underestimate peak values, as explained in the text in footnote 3. For reference, actual (non-gridded) impacts are indicated at four local maxima.

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Figure 4.5 Particle deposition velocities estimated with CARB procedures. Curves correspond to differing roughness heights (z_0). Deposition velocities reflect the distribution of meteorological observations collected at the Stewart Air Force Base.

4.2　Estimated contaminant mass fractions in soil

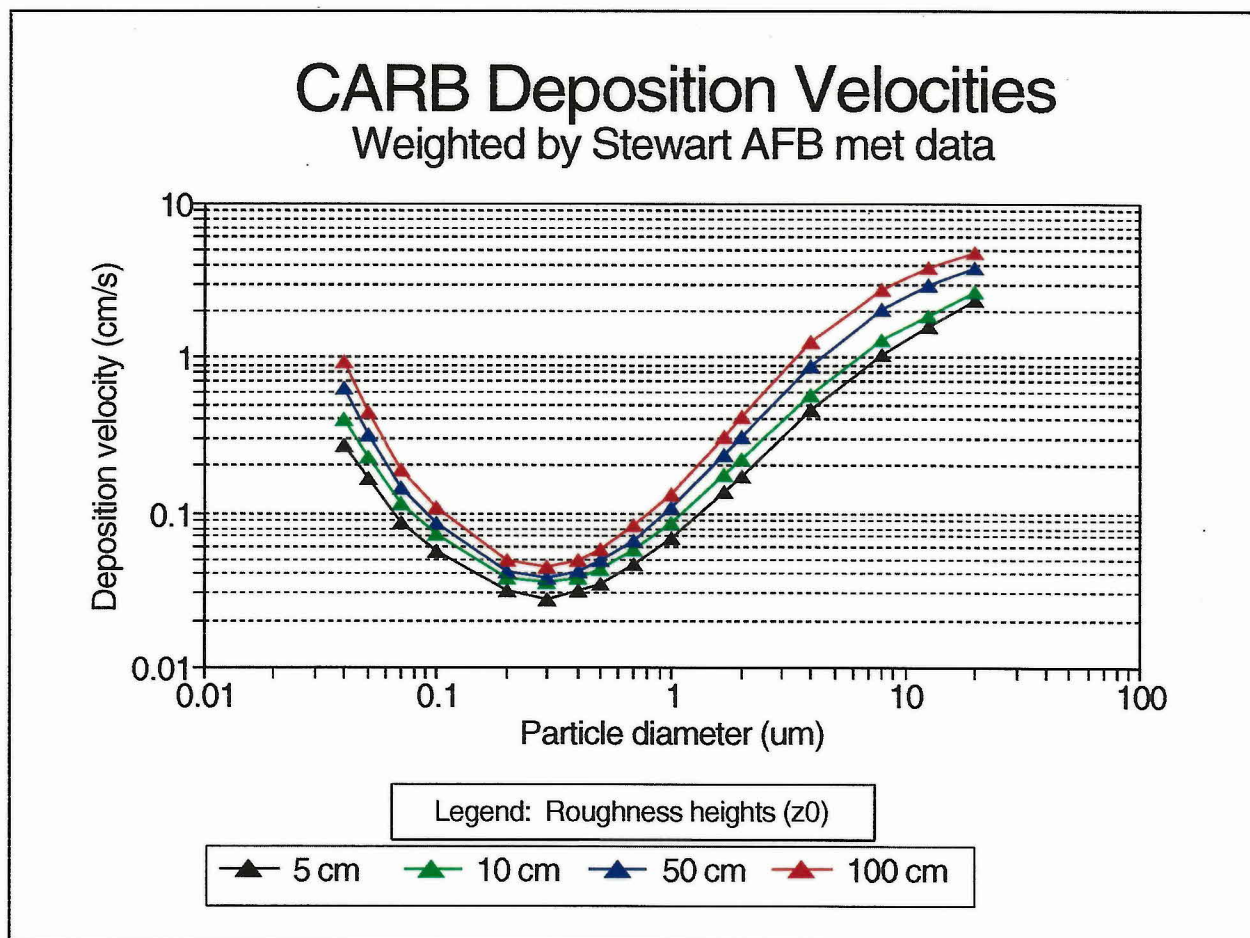Particulate-bound contaminants emitted from the NEPERA incinerator will deposit onto surface soils and surface water in the surrounding area. The contaminant mass fractions in local surface soils are estimated using the model described in NYSDOH (1991):

$$m_s = \frac{D_s \, \tau}{\rho_s \, z} \tag{4.5}$$

where the terms are

m_s　　contaminant mass fraction in soil (mg/kg),
D_s　　rate of contaminant deposition to the soil (mg/m²-yr),
τ　　time to attain average soil concentration during 70 years of plant operation (yr),
ρ_s　　soil bulk density (kg/m³), and
z　　soil mixing depth (m).

This model assumes that the contaminants reaching the ground *via* deposition will accumulate in the soils over 70 years. The average contaminant mass fraction in surface soil over 70 years of plant operation is estimated by using a τ of 35 years — the mid-point of the assumed period of operation.

Two soil mixing depths (z) are considered to account for differences in soil conditions in cultivated and uncultivated lands. Consistent with NYSDOH (1991), the tilled soil mixing depth is assumed to be 15 cm and the untilled soil mixing depth is assumed to be 5 cm. A soil bulk density (ρ_s) of 1500 kg/m³ is used as recommended by NYSDOH (1991).

Contaminant mass fractions in soil are estimated for tilled and untilled soils at the point of maximum impact (the residence of the MEI)[9]. In addition, soil mass fractions are estimated for tilled and untilled soils in the Cranberry Lake and Swimming Pond watersheds. These soil mass fractions are needed to estimate contaminant sediment loading to the water bodies, as discussed in Section 4.3. The soil mass fractions for the watersheds are calculated using the average deposition rate over the watershed areas, and thus represent the average soil mass fractions in the watersheds. Since deposition rates do not vary significantly over the watersheds, using the average soil mass fractions does not significantly affect the estimates of contaminant loading.

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[9] A site visit revealed no evidence of dairy or cattle farming at the site of maximum impact. Given no apparent barrier to such farming, however, we have assumed that the site of maximum impact will be used for dairy and cattle farming during the entire exposure period.

The deposition rates (D_s) used for calculating the soil mass fractions at the locations of interest are reported in Table 4.7. The resulting soil mass fractions are reported in Table 4.8.

4.3 Surface water concentrations

Concentrations of contaminants in surface waters, which may result from stack emissions from the NEPERA incinerator, are estimated for four lakes and ponds of potential interest to the risk assessment. Contaminant loading to each water body is assumed to occur *via* two pathways: (1) direct deposition to the surface of the water body and (2) deposition to the surface soil within the watershed followed by surface soil erosion and run-off to the water body. The total waterborne concentration (c_w) for a compound is estimated as the total mass of the compound entering the water body per unit time divided by the rate of water flow through the water body:

$$c_w = \frac{m_s R_s + D_w A_w}{V_w} \qquad (4.6)$$

where the terms are

c_w total waterborne concentration (mg/ℓ),
m_s average contaminant mass fraction in watershed run-off soils (mg/kg),
R_s sediment loading to the water body (kg/yr),
D_w deposition rate to the surface of the water body (mg/m²-yr),
A_w surface area of the water body (m²), and
V_w water volume flow rate through water body (ℓ/yr).

The water volume flow rate V_w is estimated as the total amount of precipitation recharge that enters the water body.[10] This total includes precipitation that falls directly on the water surface and the surface runoff that falls within the watershed:

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[10] This assumption neglects to account for any groundwater discharge to the waterbody. Since the addition of groundwater discharge would serve to dilute (decrease) the predicted concentration in surface water, the assumption is conservative.

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$$V_w = \alpha \left[A_w d_w + \left(A_{lw} + A_{lo} \right) d_l \right] \tag{4.7}$$

where the additional terms are

A_{lw} the area of land in the watershed covered by woods (m^2),

A_{lo} the area of cleared (open) land in the watershed (m^2),

d_w the total depth of precipitation to the water surface (m/yr),

d_l the depth of surface water runoff (m/yr), and

α a units conversion factor of 1000 ℓ/m^3.

Finally, the depth of surface water runoff d_l is estimated using an algorithm described in Mills *et al.* (1982):

$$d_l = \frac{\left(d_w - 0.2 d_s \right)^2}{d_w + 0.8 d_s}$$

$$d_s = \beta \left(\frac{1000}{CN} - 10 \right) \tag{4.8}$$

where the additional terms are

d_s the water retention parameter (m/yr),

CN the Soil Conservation Service runoff curve number, and

β a numerical constant of 0.0254 .

A value of 60 is selected for CN based upon the predominance of gravelly, silty soils in the Harriman area (USDA, 1981), wooded land-cover over the majority of the watershed areas, and the assumption of average hydrologic conditions (Mills *et al.*, 1982). We use a precipitation depth d_w of 1.21 m (47.51 inches) as provided by the Northeast Regional Climate Center (1994). The final column of Table 4.12 lists the values of V_w predicted by Equation (4.7) with the aforementioned values of CN, d_w, and watershed-specific land areas.

Estimates of total waterborne concentrations for two waterbodies used to assess exposure — Cranberry Lake and the Swimming Pond — are reported in Table 4.9 and Table 4.10, respectively. The details of the calculation and the selection of the input values follow.

Estimation of the contribution of soil erosion and runoff to the surface water concentration requires an estimate of the sediment loading (R_s) to the water body — that is, the rate at which surface soil is eroded and transported to the lake or pond. The rate of sediment loading is estimated as a function of gross soil erosion and sediment delivery using the Universal Soil Loss

Equation methodology described in Wischmeier and Smith (1978) and Mills *et al.* (1982). Estimates of net soil erosion (R_s) to each of the four water bodies of concern are reported in Table 4.12, along with values of V_w and A_w required by Equation (4.6). Calculations of R_s are discussed in detail in Section 4.3.1.

The average contaminant mass fractions in watershed soils (m_s) are calculated in Section 4.2 and reported in Table 4.8. Lands within each watershed are assumed to be untilled. The rate of direct deposition to the surface water (D_w) is calculated using air dispersion and particulate deposition models as discussed in Section 4.1. These values are reported in Table 4.7.

Model input values for calculating total pollutant concentrations in water in the Cranberry Lake and in the Swimming Pond are provided in Table 4.9 and Table 4.10.

4.3.1 Sediment loading from watershed soil erosion

Sediment loading that results from watershed soil erosion is calculated using the Universal Soil Loss Equation methodology described by Wischmeier and Smith (1978) and Mills *et al.* (1982). The watershed is divided into two land categories based upon surface cover. Wooded and open (cleared) land use categories are considered.[11] The sediment loading from each category is determined as a function of gross soil erosion and sediment delivery. The total sediment loading to the water body is simply the sum of the contributions from both categories.

The Universal Soil Loss Equation (USLE) provides an estimate of the gross soil erosion, accounting for the local terrain slope, surface cover, soil type, and meteorological conditions. Combining the USLE with the sediment delivery ratio, the fraction of gross erosion that eventually flows into the water body, the sediment loading function is

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[11] Ordinarily, agricultural lands would also be considered. Agricultural district maps, however, indicate no farming lands within the watersheds of concern.

$$R_s = \gamma \left[A_{lw} \left(C K * L_{S,avg} \right)_w + A_{lo} \left(C K * L_{S,avg} \right)_o \right] \left[R P S_d \right] \qquad (4.9)$$

where the subscripts w and o refer to wooded and open land use, respectively, and the additional terms are

R_s	reservoir sediment loading from soil erosion (tons/year),
R	rainfall factor, which expresses the erosion potential of average annual rainfall in the locality,
$K*L_{S,avg}$	average product of K and L_S factors,
K	soil-erodibility factor for a particular soil type (tons/acre-yr),
L_S	topographic factor, dependent upon slope-length and slope-steepness,
C	cover factor, dependent upon vegetation and land cover (dimensionless),
P	erosion control practice factor (dimensionless),
S_d	sediment delivery ratio (dimensionless), and
γ	units conversion factor equal to 0.0002471 acres/m².

Watershed areas were determined from topographic maps (USGS, 1981a, 1981b) by locating groundwater divides and estimating runoff travel pathways. Delineated watershed areas were traced onto transparent grids to estimate land areas. These areas are listed in Table 4.12 in units of km², and must be converted to m² (by multiplying by 1×10^6) for use in Equation (4.9).

Several of the parameters used in Equation (4.9) are assumed to be the same for all four watersheds. The rainfall factor (R), a measure of the potential for precipitation to cause erosion at a locality, is taken to be 160 based upon the geographic distribution contained in Wischmeier and Smith (1978). The practice factor (P) is used to reduce erosion estimates for lands where soil-conserving agricultural practices (e.g. contouring, contour stripcropping, and terracing) are employed to slow the erosion process. Since agricultural lands are not present in the watersheds of interest, the P factor is appropriately set to unity. The sediment delivery ratio (S_d) varies with the area of the watershed — the smaller the watershed, the higher the delivery ratio (Vanoni, 1975). Based upon this work, the following correlation developed by the U.S. EPA (1993b) is used to estimate the sediment delivery ratio:

$$S_d = 0.338 \left(A_{lw} + A_{lo} \right)^{-0.125} \qquad (4.10)$$

where the areas A_{lw} and A_{lo} are entered in units of km² (as listed in Table 4.12).

The cover factor (C) accounts for the effect of vegetation on soil erosion. Values of C for wooded and open lands are based upon typical values listed in Wischmeier and Smith (1978) and Mills et al. (1982). The C factor for wooded land is taken to be 0.006, which corresponds to the middle of the upper-range of C factors reported for undisturbed forest land. Open land in the watershed was observed to be well-covered with a low canopy. A C factor of 0.1 is selected to

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represent these lands, which is representative of a relatively high erosion protection for well-covered lands.

Values of R, S_d, C, A_{lw}, and A_{lo} are compiled in Table 4.12. Also included in Table 4.12 are weighted averages of the product of the K and L_s factors derived for each watershed. The supporting data for calculating the average $K*L_s$ values are presented in Table 4.11. These data are obtained from a soil survey conducted within Orange County (USDA, 1981), in which the geographic distribution and properties of surficial soils have been catalogued.

Of specific interest, the soil survey contains areal maps of soil types, which are identified by three letter acronyms such as those listed in Table 4.11. Using professional judgement, dominant soil types were identified within each of the four watersheds of interest, along with weighting factors to approximate the distribution of soil types within the watershed. Soil types and weighting factors (f_s) are listed in Table 4.11.

Associated with each soil type are an erodibility factor K, an average slope θ_s, and a slope-length λ_s. Values of L_s are calculated from θ and λ_s by (Wischmeier and Smith, 1978):[12]

$$L_s = \left(\frac{\lambda_s}{72.6}\right)^{0.5} \left(65.41\ \sin^2(\theta) + 4.56\ \sin(\theta) + 0.065\right)$$

(4.11)

$$\theta = \tan^{-1}\left(\frac{\theta_s}{100}\right)$$

where θ_s is a percentage value. Values of K and θ_s are taken from tables in the soil survey (USDA, 1981), and λ values are based upon similar soil types in Rensselaer County (Swanson, 1994). Within each watershed in Table 4.11, $K*L_s$ values are calculated for each soil type and weighted by the f_s fractions to produce an average value of the product of K and L_s for use in Equation (4.9):

$$K*L_{S,avg} = \sum^{n} f_s\, K*L_s$$

(4.12)

where n represents the number of soil types in the watershed. Note that the sum of f_s values used to calculate $K*L_s$ must sum to unity.

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[12] θ is related to θ_s (the percent slope) by $\theta = \tan^{-1}(\theta_s/100)$.

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A single $KL_{S,avg}$ value is developed in three of the four watersheds in which wooded lands are predominant. In the Swimming Pond watershed, however, significant fractions of both wooded and open lands are present. Consequently, separate values of $KL_{S,avg}$ are calculated for the two categories and reported in Table 4.11 and Table 4.12.

4.4 Estimated contaminant mass fractions in homegrown produce and cattle feed

Vegetation can assimilate contaminants through two mechanisms: (1) direct deposition to the exposed surfaces of plants and (2) deposition onto the soil layer followed by uptake through the root systems of plants. Both of these mechanisms are considered when estimating contaminant mass fractions in homegrown produce and animal feeds that may result from operation of the NEPERA incinerator.

4.4.1 Method for estimating contaminant mass fractions in vegetation

Contaminant mass fractions in homegrown produce and animal feeds are estimated using the methodology recommended by NYSDOH (1991). Particle-borne contaminants emitted from the NEPERA incinerator will deposit on vegetation in the surrounding area. The potential for local vegetation to intercept and incorporate settling particles depends on a variety of factors, including rate of contaminant deposition, the fraction of deposition intercepted (which depends principally upon the cross-sectional area covered by the vegetation), the rate at which degradation and weathering processes remove contaminants that have deposited, the length of the growing season, and the yield of the particular crop or vegetable. The model used to estimate contaminant mass fraction in vegetation due to atmospheric deposition takes account of these factors (NYSDOH, 1991):

$$m_{vd} = D_p^a r \frac{1 - e^{-k_v t_v}}{y k_v} \tag{4.13}$$

where the terms are

m_{vd} estimated contaminant mass fraction in produce or feed due to direct deposition of particulate matter (mg/kg dry weight or wet weight, according to the convention used for y),

D_p^a annual rate of contaminant deposition (mg/m^2-yr),

r vegetation-specific intercept fraction,

y produce or crop yield (kg/m^2),

k_v effective removal (or degradation) rate from plant surfaces (yr^{-1}), and

t_v length of the growing season (yr).

Root uptake is the second mechanism whereby plants may assimilate contaminants that originate from the NEPERA incinerator. Through deposition, particulate-bound contaminants are assumed to deposit onto the surface soils as discussed in Section 4.2. Compounds in the soil may become incorporated into vegetation *via* uptake through the root system. The degree of uptake will vary with type of the vegetation and the availability of the contaminant in the soil. The contaminant mass fraction in vegetation due to root uptake is estimated by (NYSDOH, 1991):

$$m_{vr} = m_s R_u d_w \tag{4.14}$$

where the terms are

m_{vr} estimated contaminant mass fraction in the produce or feed due to root uptake (mg/kg dry weight if d_w is omitted, wet weight if d_w is included),

m_s contaminant mass fraction in soil (mg/kg),

R_u root uptake factor [(mg/kg plant dry weight) per (mg/kg soil)], and

d_w dry- to wet-weight conversion factor (included only for produce for human consumption) [(mg/kg ww)/(mg/kg dw)].

Because rates of produce consumption by humans are typically reported on a wet-weight basis, the contaminant mass fractions in homegrown produce are calculated in terms of wet-weight. Feed consumption by cows is typically reported on a dry-weight basis. Thus, the contaminant mass fractions for feeds are calculated in terms of dry-weight.

The total contaminant mass fraction in vegetation (m_v, in mg/kg) is the sum of the components due to root uptake (m_{vr}) and direct deposition (m_{vd}):

$$m_v = m_{vr} + m_{vd} \tag{4.15}$$

4.4.2 Parameter values for estimating contaminant mass fractions in homegrown produce and cattle feed

A number of the parameters used in Equations (4.13) and (4.14) will vary with the particular vegetable, fruit, or crop of interest. In theory, the contaminant mass fractions in each type of produce or feed crop would be evaluated individually. A paucity of empirical data, however, necessitates the categorization of the assortment of vegetables, fruits, and crop feeds into a limited number of groups.

Based on the data available in the literature and draft guidance (NYSDOH, 1991), produce grown for human consumption is divided into three categories: leafy produce, exposed produce, and protected produce. Leafy produce (*e.g.* spinach, broccoli, and lettuce) is characterized by the

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relatively large interception fractions of their edible portion, which qualitatively suggests that — in terms of potential for exposure — direct deposition should be of greater importance relative to non-leafy vegetation. Exposed produce (*e.g.* tomatoes, bell peppers, and strawberries) includes non-leafy fruits and vegetables for which the edible portion is grown above-ground. Last, protected produce (*e.g.* oranges, carrots, and potatoes) includes fruits and vegetables that are not exposed to direct deposition — the edible portion is found below ground or protected by an inedible casing.

In accordance with NYSDOH guidance, beef and dairy cattle are assumed to consume a mixture of hay, corn silage, grain, and pasture grass along with a small amount of soil — incidental soil ingestion may occur during grazing. The potential contaminant mass fractions in hay, corn silage, grain and pasture grass are evaluated at the point of maximum impact.

The contaminant deposition rates and soil mass fractions are reported in Table 4.7 and Table 4.8, respectively. The empirical parameters (r, y, k_v, t_v, d_w, and R_u) used to calculate the contaminant mass fractions in homegrown produce and feed are summarized in Table 4.13. The root uptake factors (R_u) are specific to compound and vegetation type. The values of R_u are taken from NYSDOH (1991) guidance where available, or from Baes *et al.* (1984). For homegrown produce (leafy, exposed, and protected), grain, and pasture grass, the parameter values for (r, y, k_v, t_v, and d_w) are consistent with NYSDOH (1991) guidance. For the hay and corn silage, NYSDOH guidance values are used for the washoff coefficient (k_v), while information on site-specific agricultural practices and statistics is used to determine the average growing times (t_v), yields (y), and intercept fraction (r). According to the Cornell Cooperative Extension Office for Orange County, the average growing times for hay and corn silage in this region are 40 days and 90 days, respectively (Hull, 1994). Hay and corn silage yields[13] are reported for Orange County for 1989-1990 (NYASS, 1991; NYASS, 1993). For this period, the average yields[14] (kg dry weight/m^2) for hay and corn silage in this area were 0.14 and 0.87. The intercept fractions (r) for hay and corn silage are calculated using the relationships between yield

---

[13] For hay, the yields reported in NYASS (1991, 1993) are total yield per acre over the farming season. Since hay is harvested every 5 to 6 weeks, a typical farmer in the region would make about 3 cuttings of hay in a season; thus, the yield per cutting is estimated as one third of the total yield.

[14] The yields reported in NYASS (1991, 1993) are not on a dry-weight basis. According to the New York Agricultural Statistics Service, the reported yields are based on weight at harvest. Typically, a crop is cut and allowed to dry in the field some before collection. The reported yields are converted to a dry-weight basis using the semi-dry to dry-weight conversion factors for hay and corn silage of 0.897 and 0.341, provided in Hoffman and Baes (1979).

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and intercept fraction reported in Baes *et al.* (1984) for silage and hay (assumed to be similar to pasture grass):

$$r = 1 - e^{-0.769\,y} \quad (\text{ corn silage })$$
$$r = 1 - e^{-2.88\,y} \quad (\text{ hay })$$

(4.16)

The estimated contaminant mass fractions in leafy, exposed, and protected produce grown at the MEI residence are reported in Table 4.14. The contaminant mass fractions estimated for hay, corn silage, grain, and pasture grass grown at the point of maximum impact are reported in Table 4.15. In order to allow the reader to discern the relative importance of the deposition and uptake processes, the fraction of the total mass fraction due to direct deposition is also provided in the tables.

4.5 Estimated contaminant mass fractions in beef and dairy products

For compounds that bioconcentrate, the mass fractions in beef from cattle or milk from dairy cows may be related to the total daily intake of the compounds by these animals. In accordance with NYSDOH (1991) guidance, the contaminant mass fractions in beef (m_b) and milk (m_m) are related to dietary intake (I_d) by biotransfer factors (F_b and F_m for beef and milk, respectively):

$$m_b = F_b I_d \qquad \text{beef}$$

(4.17)

and

$$m_m = F_m I_d \qquad \text{milk}$$

(4.18)

where the terms are

m_b and m_m contaminant mass fractions (mg/kg) in beef and milk, respectively,
I_d daily contaminant intake by the animal (mg/day), and
F_b and F_m biotransfer factors (days/kg) for beef and milk, respectively.

As discussed in Section 4.4.2, dietary parameters recommended by NYSDOH (1991) are used for the evaluation of beef cattle and dairy cows feed intakes at the location of the MEI receptor.

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The dietary intake of the animal is simply the sum of contaminants derived from each source of feed. The dietary intake is calculated as:

$$I_d = R_h m_h + R_c m_c + R_g m_g + R_p m_p + R_s m_s \tag{4.19}$$

where the terms are

R_h, R_c, R_g, R_p, and R_s ingestion rates of hay, corn silage, grain, pasture grass, and soil (kg dry weight/day), and

m_h, m_c, m_g, m_p, and m_s contaminant mass fractions in hay, corn silage, grain, pasture grass, and soil (mg/kg dry weight).

Dietary ingestion rates differ for lactating cows and non-lactating cows and cattle. NYSDOH (1991) guidance provides feed consumption rates for dairy cows and beef cattle, splitting the total intake among hay, pasture grass, corn silage, and grain. These intake rates are provided in Table 4.16. The contaminant mass fractions in hay, corn silage, grain, and pasture grass are calculated in Section 4.4 and reported in Table 4.15. The contaminant mass fractions in soil are calculated in Section 4.2 and reported in Table 4.8. The beef and milk biotransfer factors, F_b and F_m, are taken from NYSDOH (1991) guidance where available, or from Baes *et al.* (1984). The resulting contaminant mass fractions in beef and milk produced at the most-impacted farm are also reported in Table 4.17.

4.6   Estimated contaminant mass fractions in fish

4.6.1   Method for estimating the contaminant mass fractions in fish

The contaminant mass fractions in fish (m_f) are assumed to be proportional to contaminant concentrations in water for all chemicals of potential concern. The mass fraction in fish m_f (in mg/kg) is estimated by:

$$m_f = B_{cf} c_w \tag{4.20}$$

where (c_w in mg/l) is the pollutant concentration in water and B_{cf} is the bioconcentration factor (in l/kg). Surface water concentrations are estimated in Section 4.3. The surface water concentrations estimated for Cranberry Lake, the lake of greatest potential concern for fishing, are repeated in Table 4.18, along with bioconcentration factors (B_{cf}) and the subsequently estimated mass fractions of pollutants in fish (m_f).

Table 4.1   Stack and Flue Gas Parameters

Model parameter	Assumed value
Stack height	46.3 m (152 ft)
Stack elevation above mean sea level (amsl)	162 m (531.5 ft)
Stack exit diameter	1.5 m
Stack gas temperature	603 K
Stack gas exit velocity	14.3 m/s

Table 4.2   Modeled Air Pollutant Concentrations at Ponds and Lakes

Lake or Pond	Potential exposure routes	Modeled air pollutant concentration (μg/m^3 per g/s)	
		To water surface	To watershed lands
Blendale, Blythea, & Shadowmere Lakes	Fishing	0.98	0.89
Sapphire Lake	Fishing	0.96	0.76
Swimming Pond (Monroe)	Swimming	0.18	0.73
Cranberry Lake	Drinking water ingestion & fishing	0.72	0.70

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Table 4.3   Supporting data used to estimate air dispersion modeling impacts at surface water bodies

Watershed	Polar receptor coordinates		Modeled air concentration at receptor (α_c) ($\mu g/m^3$ per g/s)	Weighting factors (f_c)		Averaged (weighted) concentration in air	
	Radius from stack	Angle (from north)		Water	Land	Water	Land
Blendale, Blythea, & Shadowmere Lakes	2250	230	1.09	0.5	0.5	0.98	0.89
	2250	220	1.17	0.5	0.5		
	2500	230	1.01	0	0.5		
	2500	220	1.02	0	0.5		
	2750	230	0.90	1	1		
	2750	220	0.90	1	1		
	3000	230	0.80	0	1		
	3000	220	0.79	0	1		
	3250	230	0.71	0	0.5		
	3250	220	0.67	0	0.5		
Sapphire Lake	2500	210	1.02	1	0.5	0.96	0.76
	2750	210	0.90	1	1		
	3000	210	0.79	0	1		
	3250	210	0.67	0	1		
	3500	210	0.60	0	0.5		
	3750	210	0.47	0	0.5		

Table 4.3   Supporting data used to estimate air dispersion modeling impacts at surface water bodies

Watershed	Polar receptor coordinates		Modeled air concentration at receptor (α_c) ($\mu g/m^3$ per g/s)	Weighting factors (f_c)		Averaged (weighted) concentration in air	
	Radius from stack	Angle (from north)		Water	Land	Water	Land
Swimming Pond (Monroe)	2000	260	1.10	0	0.5	0.18	0.73
	2250	260	0.65	0	0.5		
	2500	260	0.46	0	0.5		
	2750	260	0.18	1	0		
	2000	250	0.91	0	0.5		
	2250	250	1.09	0	0.5		
	2500	250	0.84	0	1		
	2750	250	0.43	0	1		
	3000	250	0.68	0	1		
	3250	250	0.69	0	0.5		
	3250	240	0.71	0	0.5		
Cranberry Lake	2750	120	0.75	0	1	0.72	0.70
	3000	120	0.72	1	1		
	3250	120	0.65	0	1		
	2750	130	0.81	0	1		
	3000	130	0.72	1	1		
	3250	130	0.60	0	1		
	3000	140	0.64	0	1		

Table 4.4   Emission rates for the NEPERA incinerator

Contaminants	Emission rate (g/s)
Ammonia	1.32e-02
Antimony	1.83e-04
Arsenic	1.29e-05
Barium	3.52e-04
Beryllium	7.11e-07
Cadmium	7.76e-06
Chromium VI	8.93e-05
Chromium (total)	5.63e-04
Copper	3.54e-04
Lead	3.27e-04
Manganese	1.13e-03
Mercury	8.06e-05
Nickel	1.49e-03
Selenium	7.22e-06
Silver	1.31e-05
Thallium	1.69e-06
Vanadium	1.98e-05
Zinc	1.98e-03
Alkyl pyridine	2.16e-02
2-Cyanopyridine	6.42e-03
3-Cyanopyridine	6.42e-03
Benzene	4.38e-03
Ethanol	3.41e-03
Isopropanol	3.41e-03
Methanol	3.41e-03
2-Picoline	5.47e-03
3-Picoline	5.48e-03
Pyridine	3.44e-02
Toluene	6.81e-04

Table 4.5 Air concentrations at point of maximum impact

Pollutant	Air concentration ($\mu g/m^3$)
Ammonia	1.78e-02
Antimony	2.47e-04
Arsenic	1.75e-05
Barium	4.75e-04
Beryllium	9.60e-07
Cadmium	1.05e-05
Chromium VI	1.21e-04
Chromium (total)	7.60e-04
Copper	4.78e-04
Lead	4.42e-04
Manganese	1.52e-03
Mercury	1.09e-04
Nickel	2.01e-03
Selenium	9.75e-06
Silver	1.77e-05
Thallium	2.28e-06
Vanadium	2.67e-05
Zinc	2.67e-03
Alkyl pyridine	2.91e-02
2-Cyanopyridine	8.67e-03
3-Cyanopyridine	8.67e-03
Benzene	5.91e-03
Ethanol	4.60e-03
Isopropanol	4.60e-03
Methanol	4.60e-03
2-Picoline	7.38e-03
3-Picoline	7.39e-03
Pyridine	4.64e-02
Toluene	9.20e-04

Table 4.6   Normalized deposition rates

Location		R (m)	Theta	Deposition Rates (g/m²-s)/(g/s)	
				Mass-weighted	Area-weighted
MEI Residence		2000	120	1.35e-09	1.35e-09
Blendale, Blythea, & Shadowmere Lakes	deposition to soil	average over watershed		8.90e-10	8.90e-10
	deposition to water	average over water		9.80e-10	9.80e-10
Sapphire Lake	deposition to soil	average over watershed		7.60e-10	7.60e-10
	deposition to water	average over water		9.60e-10	9.60e-10
Swimming Pond	deposition to soil	average over watershed		7.30e-10	7.30e-10
	deposition to water	average over water		1.80e-10	1.80e-10
Cranberry Lake	deposition to soil	average over watershed		7.00e-10	7.00e-10
	deposition to water	average over water		7.20e-10	7.20e-10

Table 4.7 Deposition rates

Contaminant	Mass-weighted (MW) or Surface-weighted (SW)	Deposition rates (g/m²/s)[a]				
		MEI residence	Cranberry Lake		Swimming Pond	
			soil	water	soil	water
Ammonia	NA	NA	NA	NA	NA	NA
Antimony	MW	2.47e-13	1.28e-13	1.32e-13	1.34e-13	3.29e-14
Arsenic	MW	1.75e-14	9.06e-15	9.32e-15	9.45e-15	2.33e-15
Barium	MW	4.75e-13	2.46e-13	2.54e-13	2.57e-13	6.34e-14
Beryllium	MW	9.60e-16	4.98e-16	5.12e-16	5.19e-16	1.28e-16
Cadmium	MW	1.05e-14	5.43e-15	5.59e-15	5.66e-15	1.40e-15
Chromium VI	MW	1.21e-13	6.25e-14	6.43e-14	6.52e-14	1.61e-14
Chromium (total)	MW	7.60e-13	3.94e-13	4.05e-13	4.11e-13	1.01e-13
Copper	MW	4.78e-13	2.48e-13	2.55e-13	2.58e-13	6.37e-14
Lead	SW	4.42e-13	2.29e-13	2.36e-13	2.39e-13	5.89e-14
Manganese	MW	1.52e-12	7.91e-13	8.13e-13	8.24e-13	2.03e-13
Mercury	SW	1.09e-13	5.64e-14	5.80e-14	5.89e-14	1.45e-14
Nickel	MW	2.01e-12	1.04e-12	1.07e-12	1.09e-12	2.68e-13
Selenium	SW	9.75e-15	5.06e-15	5.20e-15	5.27e-15	1.30e-15
Silver	MW	1.77e-14	9.19e-15	9.45e-15	9.58e-15	2.36e-15
Thallium	MW	2.28e-15	1.18e-15	1.22e-15	1.23e-15	3.04e-16
Vanadium	MW	2.67e-14	1.39e-14	1.43e-14	1.45e-14	3.56e-15
Zinc	MW	2.67e-12	1.38e-12	1.42e-12	1.44e-12	3.56e-13
Alkyl pyridine	NA	NA	NA	NA	NA	NA
2-Cyanopyridine	NA	NA	NA	NA	NA	NA
3-Cyanopyridine	NA	NA	NA	NA	NA	NA
Benzene	NA	NA	NA	NA	NA	NA
Ethanol	NA	NA	NA	NA	NA	NA
Isopropanol	NA	NA	NA	NA	NA	NA
Methanol	NA	NA	NA	NA	NA	NA
2-Picoline	NA	NA	NA	NA	NA	NA
3-Picoline	NA	NA	NA	NA	NA	NA
Pyridine	NA	NA	NA	NA	NA	NA
Toluene	NA	NA	NA	NA	NA	NA

NA = Not applicable

[a] Deposition rates are reported only for those water bodies used for the exposure assessment. Of the waterbodies evaluated, Cranberry Lake is the most-impacted and is used as the source for drinking water and fish. The Swimming Pond is used to evaluate exposure from swimming.

Table 4.8   Soil mass fractions

Contaminant	Soil mass fractions (mg/kg)[a]					
	MEI residence		Cranberry Lake		Swimming Pond	
	Untilled	Tilled	Untilled	Tilled	Untilled	Tilled
Ammonia	NA	NA	NA	NA	NA	NA
Antimony	3.64e-03	1.21e-03	1.89e-03	6.29e-04	1.97e-03	6.56e-04
Arsenic	2.57e-04	8.58e-05	1.33e-04	4.45e-05	1.39e-04	4.64e-05
Barium	7.00e-03	2.33e-03	3.63e-03	1.21e-03	3.79e-03	1.26e-03
Beryllium	1.41e-05	4.71e-06	7.33e-06	2.44e-06	7.64e-06	2.55e-06
Cadmium	1.54e-04	5.14e-05	8.00e-05	2.67e-05	8.34e-05	2.78e-05
Chromium VI	1.78e-03	5.92e-04	9.21e-04	3.07e-04	9.60e-04	3.20e-04
Chromium (total)	1.12e-02	3.73e-03	5.80e-03	1.93e-03	6.05e-03	2.02e-03
Copper	7.03e-03	2.34e-03	3.65e-03	1.22e-03	3.80e-03	1.27e-03
Lead	6.51e-03	2.17e-03	3.38e-03	1.13e-03	3.52e-03	1.17e-03
Manganese	2.25e-02	7.48e-03	1.16e-02	3.88e-03	1.21e-02	4.05e-03
Mercury	1.60e-03	5.34e-04	8.31e-04	2.77e-04	8.67e-04	2.89e-04
Nickel	2.97e-02	9.88e-03	1.54e-02	5.13e-03	1.60e-02	5.34e-03
Selenium	1.44e-04	4.79e-05	7.45e-05	2.48e-05	7.77e-05	2.59e-05
Silver	2.61e-04	8.70e-05	1.35e-04	4.51e-05	1.41e-04	4.70e-05
Thallium	3.36e-05	1.12e-05	1.74e-05	5.80e-06	1.82e-05	6.05e-06
Vanadium	3.94e-04	1.31e-04	2.04e-04	6.81e-05	2.13e-04	7.10e-05
Zinc	3.93e-02	1.31e-02	2.04e-02	6.79e-03	2.13e-02	7.09e-03
Alkyl pyridine	NA	NA	NA	NA	NA	NA
2-Cyanopyridine	NA	NA	NA	NA	NA	NA
3-Cyanopyridine	NA	NA	NA	NA	NA	NA
Benzene	NA	NA	NA	NA	NA	NA
Ethanol	NA	NA	NA	NA	NA	NA
Isopropanol	NA	NA	NA	NA	NA	NA
Methanol	NA	NA	NA	NA	NA	NA
2-Picoline	NA	NA	NA	NA	NA	NA
3-Picoline	NA	NA	NA	NA	NA	NA
Pyridine	NA	NA	NA	NA	NA	NA
Toluene	NA	NA	NA	NA	NA	NA

NA = Not applicable

[a] Average soil concentrations are reported the watersheds of those water bodies used for the exposure assessment. Of the waterbodies evaluated, Cranberry Lake is the most-impacted and is used as the source for drinking water and fish. The Swimming Pond is used to evaluate exposure from swimming.

Table 4.9   Total waterborne concentrations in Cranberry Lake

Contaminant	Soil mass fraction m_s (mg/kg)	Soil loading R_s (kg/yr)	Deposition rate to water D_w (mg/m²-yr)	Surface area of water A_w (m²)	Volume flow rate V_w (l/yr)	Concentration in water c_w (mg/l)
Ammonia	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
Antimony	1.89e-03	6.80e+04	4.16e-03	3.13e+05	9.85e+08	1.45e-06
Arsenic	1.33e-04	6.80e+04	2.94e-04	3.13e+05	9.85e+08	1.03e-07
Barium	3.63e-03	6.80e+04	8.00e-03	3.13e+05	9.85e+08	2.80e-06
Beryllium	7.33e-06	6.80e+04	1.62e-05	3.13e+05	9.85e+08	5.65e-09
Cadmium	8.00e-05	6.80e+04	1.76e-04	3.13e+05	9.85e+08	6.16e-08
Chromium VI	9.21e-04	6.80e+04	2.03e-03	3.13e+05	9.85e+08	7.10e-07
Chromium (total)	5.80e-03	6.80e+04	1.28e-02	3.13e+05	9.85e+08	4.47e-06
Copper	3.65e-03	6.80e+04	8.04e-03	3.13e+05	9.85e+08	2.81e-06
Lead	3.38e-03	6.80e+04	7.44e-03	3.13e+05	9.85e+08	2.60e-06
Manganese	1.16e-02	6.80e+04	2.57e-02	3.13e+05	9.85e+08	8.97e-06
Mercury	8.31e-04	6.80e+04	1.83e-03	3.13e+05	9.85e+08	6.41e-07
Nickel	1.54e-02	6.80e+04	3.39e-02	3.13e+05	9.85e+08	1.19e-05
Selenium	7.45e-05	6.80e+04	1.64e-04	3.13e+05	9.85e+08	5.74e-08
Silver	1.35e-04	6.80e+04	2.98e-04	3.13e+05	9.85e+08	1.04e-07
Thallium	1.74e-05	6.80e+04	3.84e-05	3.13e+05	9.85e+08	1.34e-08
Vanadium	2.04e-04	6.80e+04	4.50e-04	3.13e+05	9.85e+08	1.57e-07
Zinc	2.04e-02	6.80e+04	4.49e-02	3.13e+05	9.85e+08	1.57e-05
Alkyl pyridine	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
2-Cyanopyridine	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
3-Cyanopyridine	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
Benzene	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
Ethanol	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
Isopropanol	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
Methanol	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
2-Picoline	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
3-Picoline	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
Pyridine	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA
Toluene	NA	6.80e+04	NA	3.13e+05	9.85e+08	NA

NA = Not applicable

Table 4.10   Total waterborne concentrations in the Swimming Pond

Contaminant	Soil mass fraction m_s (mg/kg)	Soil loading R_s (kg/yr)	Deposition rate to water D_w (mg/m²-yr)	Surface area of water A_w (m²)	Volume flow rate V_w (l/yr)	Concentration in water c_w (mg/l)
Ammonia	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
Antimony	1.97e-03	3.93e+05	1.04e-03	4.23e+04	7.53e+08	1.08e-06
Arsenic	1.39e-04	3.93e+05	7.36e-05	4.23e+04	7.53e+08	7.67e-08
Barium	3.79e-03	3.93e+05	2.00e-03	4.23e+04	7.53e+08	2.09e-06
Beryllium	7.64e-06	3.93e+05	4.04e-06	4.23e+04	7.53e+08	4.21e-09
Cadmium	8.34e-05	3.93e+05	4.41e-05	4.23e+04	7.53e+08	4.60e-08
Chromium VI	9.60e-04	3.93e+05	5.07e-04	4.23e+04	7.53e+08	5.29e-07
Chromium (total)	6.05e-03	3.93e+05	3.20e-03	4.23e+04	7.53e+08	3.34e-06
Copper	3.80e-03	3.93e+05	2.01e-03	4.23e+04	7.53e+08	2.10e-06
Lead	3.52e-03	3.93e+05	1.86e-03	4.23e+04	7.53e+08	1.94e-06
Manganese	1.21e-02	3.93e+05	6.41e-03	4.23e+04	7.53e+08	6.69e-06
Mercury	8.67e-04	3.93e+05	4.58e-04	4.23e+04	7.53e+08	4.78e-07
Nickel	1.60e-02	3.93e+05	8.47e-03	4.23e+04	7.53e+08	8.84e-06
Selenium	7.77e-05	3.93e+05	4.10e-05	4.23e+04	7.53e+08	4.28e-08
Silver	1.41e-04	3.93e+05	7.45e-05	4.23e+04	7.53e+08	7.78e-08
Thallium	1.82e-05	3.93e+05	9.59e-06	4.23e+04	7.53e+08	1.00e-08
Vanadium	2.13e-04	3.93e+05	1.12e-04	4.23e+04	7.53e+08	1.17e-07
Zinc	2.13e-02	3.93e+05	1.12e-02	4.23e+04	7.53e+08	1.17e-05
Alkyl pyridine	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
2-Cyanopyridine	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
3-Cyanopyridine	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
Benzene	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
Ethanol	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
Isopropanol	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
Methanol	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
2-Picoline	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
3-Picoline	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
Pyridine	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA
Toluene	NA	3.93e+05	NA	4.23e+04	7.53e+08	NA

NA = Not applicable

Table 4.11   Parameters used to estimate K^*L_S factors for use in the Universal Soil Loss Equation

Watershed	Dominant soil types in watershed		Weighting factor f_{LS}	K factor	Average slope θ_s (%)	Average length λ	L_s factor for soil type	K^*L_s factor for soil type
	Soil acronym	Description						
Blendale, Blythea, & Shadowmere Lakes	MdB	gravelly silty loam	0.2	0.24	5.5	200	0.85	0.20
	HLC	gravelly loam	0.2	0.24	11.5	150	2.08	0.50
	SXC	stony	0.2	0.17	11.5	150	2.08	0.35
	ROC	rock outcrop/HLC	0.2	0.2	11.5	150	2.08	0.42
	MdD	gravelly silty loam	0.1	0.24	20	125	4.66	1.12
	MdC	gravelly silty loam	0.1	0.24	11.5	150	2.08	0.50
	Weighted-average $K^*L_{S.avg}$ factor for both wooded and open areas							0.46
Sapphire Lake	HLC	gravelly loam	0.40	0.24	11.5	150	2.08	0.50
	ROC	rock outcrop/HLC	0.40	0.2	11.5	150	2.08	0.42
	ROD	rock outcrop/HLC	0.20	0.2	20	125	4.66	0.93
	Weighted-average $K^*L_{S.avg}$ factor for both wooded and open areas							0.55
Swimming Pond (Monroe)	HLC	gravelly loam	0.25	0.24	11.5	150	2.08	0.50
	HLD	gravelly loam	0.25	0.24	20	125	4.66	1.12
	MdC	gravelly silty loam	0.25	0.24	11.5	150	2.08	0.50
	MdB	gravelly silty loam	0.25	0.24	5.5	200	0.85	0.20
	Weighted-average $K^*L_{S.avg}$ factor for wooded areas							0.58
	MdD	gravelly silty loam	0.33	0.24	20	125	4.66	1.12
	HLC	gravelly loam	0.33	0.24	11.5	150	2.08	0.50
	ErB	gravelly silt loam	0.33	0.24	5.5	200	0.85	0.20
	Weighted-average $K^*L_{S.avg}$ factor for open areas							0.61